Ecological Economics 167 (2020) 106453

Contents lists available at ScienceDirect

Ecological Economics

journal homepage: www.elsevier.com/locate/ecolecon

Analysis

Cost-benefit analysis of conservation policy: The red palm weevil in Catalonia, Spain

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ARTICLE INFO

Keywords:
Environmental valuation
Phoenix. spp
Ecosystem services
Invasive species

ABSTRACT

Invasive species are costly for human health, the environment and the economy while their burden is expected to rise. With limited budgets to address biological invasions, effective resource allocation is important. In the past decade, multiple frameworks have emerged to support this budgeting, but it is not clear if current strategies are consistent with these. Amongst invasive species, insects are the costliest. In this article we evaluate a set of conservation policies in response to the arrival of the invasive beetle, the red palm weevil (*Rhynchophorus ferrugineus*) in Catalonia, Spain. The purpose of the selected schemes was to preserve palm species (*Phoenix.* spp) serving ornamental purposes. In a region with a large portion of land dedicated to agricultural activities and with densely populated coastal areas, budgets to address biological invasions should be carefully allocated. Through a comprehensive cost-benefit analysis based on the total economic value framework, we find that current policies were not justified as their net social benefits are negative.

1. Introduction

The introduction of invasive alien species (IAS) can be considered to be an important feature of the Anthropocene (Capinha et al., 2015; Lewis and Maslin, 2015). A worldwide growth in trade and transport during the past two centuries has increased the rate at which new IAS appear (Costello et al., 2007; Pyšek and Richardson, 2010). Depending on the taxonomic classification, 1–16% of all species on Earth qualify as potential alien species. For most taxonomic groups, the greater part of potential invasions has yet to occur (Seebens et al., 2018).

IAS often have a negative impact on human health, the economy and ecosystems (Bradshaw et al., 2016; Colautti et al., 2006; Paini et al., 2016; Pimentel et al., 2005; Vazquez-Prokopec et al., 2010; Williams et al., 2010). Invasive insects (hereinafter II) are likely to be the costliest living class to humans with an annual minimum cost estimation of USD 70 billion globally (Bradshaw et al., 2016). There are about 1.84 to 2.57 million insect species (Mora et al., 2011) and to date, only an approximate 24% of potential insect invasions have taken place (Seebens et al., 2018). Depending on the policy implemented to lower the probability of these invasions, the cost is typically assumed by taxpayers, private entities, and residential property owners (Fenichel et al., 2014; Funk et al., 2014; Lovett et al., 2016; Martelli et al., 2015; Van Damme et al., 2004).

To minimize costs borne by II, often the best policies are preventive which include measures in exporting countries and inspections of shipments at ports of entry (Lovett et al., 2016. In instances of effective II introduction, governments can apply post-entry measures such as quarantines, surveillance and eradication programs (Keller and Perrings, 2011; Thompson Campbell and Schlarbaum, 1994). However, when budgets to address biological invasions are limited and invasions are rising, what should governments and individuals do? Prioritization of which species, pathways, or sites to address is key (McGeoch et al., 2016), as stated in Target 9 of the Aichi Biodiversity Targets of Convention on Biological Diversity (SCBD, 2010).

In this article we explore a set of conservation policies designed to address the arrival of an invasive insect. This assessment supports policy makers at a local, regional and international level to improve resource allocation by considering the environmental and possibly socio-economic impacts (Blackburn et al., 2014; Kumschick et al., 2012) and choosing actions that best avoid or lessen the impact of invasive species (McGeoch et al., 2016).

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https://doi.org/10.1016/j.ecolecon.2019.106453
Received 29 November 2018; Received in revised form 20 August 2019; Accepted 26 August 2019
Available online 25 October 2019
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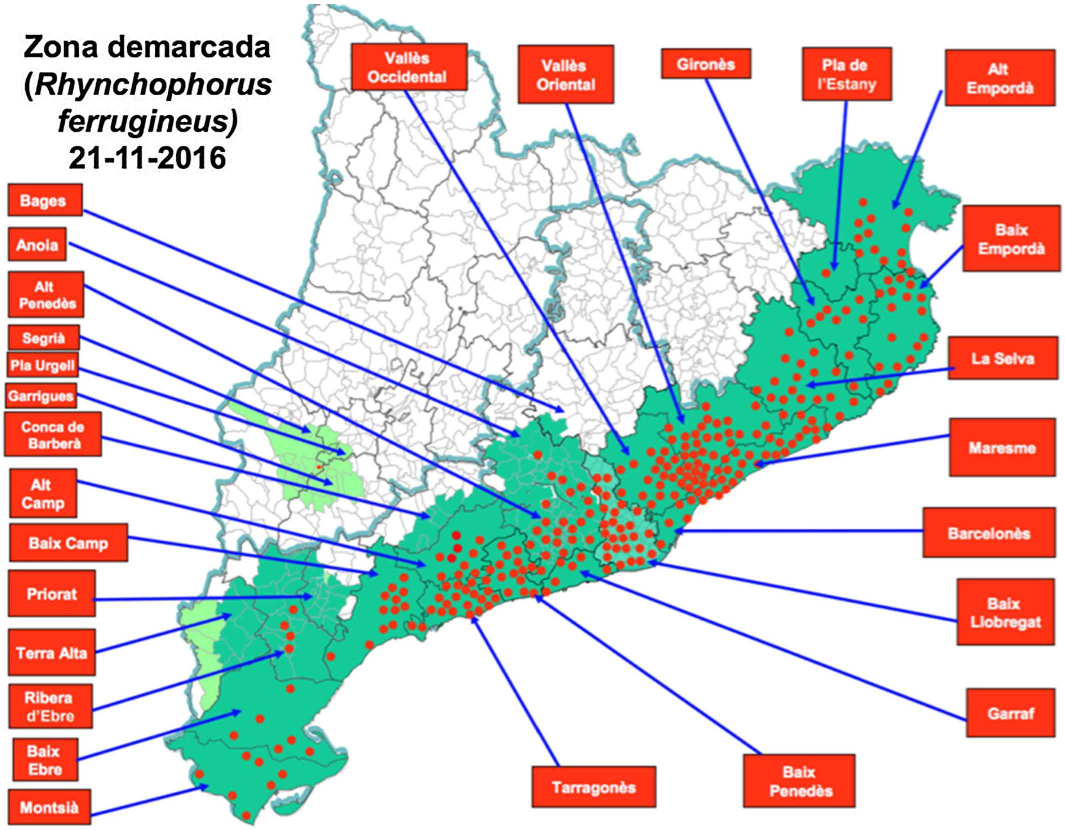


Fig. 1. RPW invasion status in 2016, the invaded area reported by GENCAT is coastal and for the most part, overlaps spatial population distribution. .
Source: Source: GENCAT (2017)

The conservation policies selected[1] . were designed to preserve two highly valued plant species: *Phoenix dactylifera* (the Date palm) and *Phoenix canariensis* (the Canary palm), both non-native and considered of 'least concern' by the IUCN Global Species Programme Red List Unit (Beech, 2017) but susceptible to an II in Catalonia, Spain. These two species have come under pressure in the region of study due to the arrival of the red palm weevil (*Rhynchophorus ferrugineus*, Coleoptera: Dryophthoridae) or as it is known locally, the 'morrut de les palmeres' (Catalan for palm weevil). II such as the red palm weevil (RPW) are introduced through trade and transport (Costello et al., 2007; Margolis et al., 2005; Meyerson and Mooney, 2007). Merchandise imports (Levine and D'Antonio, 2003) and economic and demographic variables have been found to influence invasion levels (Pysek et al., 2010).

There is a negative externality involved here as contamination of one tree with RPW poses a risk for nearby trees. Hence, there is a negative spillover effect between trees and thus also between owners of trees. This underpins the need for public policy to control tree infestation. Specific policies have been implemented in Catalonia to mitigate the impact on susceptible palm species caused by the RPW. We will use a combination of dynamic modeling and a cost-benefit analysis to assess the selected policies. In particular, we will calculate the costs and benefits associated with the policy and the scenario without such policy to accurately evaluate the net social benefit of the conservation policy. This approach has the advantage of replacing expert judgments about future populations, costs and benefits by estimates generated with a systems model that combines relevant dynamic processes which are hard to handle, let alone to quantify, intuitively (Courtois et al., 2018).

[1] The selected policies are formulated in Ordre ARP/343/2006, Ordre AAR/226/2009, Ordre AAR/2802/2010, and Ordre AAM/56/2011. These are explained in further detail in Appendix A.

2. The RPW around the world and in Catalonia

The RPW is native to Southeast Asia and Polynesia (Wattanapongsiri, 1966) being essentially a pest of palms (Arecaceae), with 29 species listed as host plants (CABI, 2017). It has been suggested to also harm sugarcane, *Saccharum officinarum*, (Poaceae) and sentry plants or maguey, *Agave americana* (Agavaceae) (CABI, 2017) although this still awaits rigorous confirmation. The RPW was introduced in other parts of the world mainly through imports of palm trees or offshoots from native areas. For instance, it reached Egypt in 1992 from the United Arab Emirates, and Spain in 1995 from imported Egyptian palms (Ferry and Gómez, 2002).

The RPW lays eggs mostly on the crown of palm trees. Larvae then penetrate it eventually reaching the crown core where the meristematic tissue resides. Subsequent feeding on this tissue will eventually kill the palm tree. Because the whole larval stage occurs within the palm tree, it is concealed from plain eyesight and hence hard to detect.

In Spain, palm damage by this beetle was first observed in 1993 on the coastal front of the Granada province (Barranco et al., 1996). In 2004 it reached the Autonomous Community of Valencia (EPPO, 2008). It was officially declared a pest in Catalonia in 2006 (Generalitat de Catalunya, 2006) where it presumably arrived between 2003 and 2004 to the Ebro river delta and Tarragona region (South Catalonia). Fig. 1 shows the invasion status of RPW in Catalonia up to November 2016 (Generalitat de Catalunya, 2017).

Several other areas in the Mediterranean have been colonized by the pest. Sicily, for instance, was reached in 2005. Here palm damage was categorised as a threat to cultural heritage (Manachini et al., 2013; Peri et al., 2013). Other countries where it appeared for the first time in this period include Turkey (2005), Cyprus (2006), Greece (2006) and Malta (2007) (Mizzi et al., 2009; Yuezhong et al., 2009). The beetle has also invaded other parts of the world. In Japan, it was first observed in 1975

Table 1
Gardens, nurseries and plant traders in Catalonia per Administrative Region. Source: GENCAT (2017).

Region	With palms	All	Ratio
Barcelona	59	348	0.17
Girona	63	162	0.39
Lleida	10	148	0.07
Tarragona	36	98	0.37
Terres de l'Ebre	37	106	0.35
Total	205	862	0.24

in the island of Okinawa, and in 1998 it spread northwards to the island of Kyushu (Abe et al., 2009). During the 1990s, the RPW was intercepted on several occasions in China. In 2007, several cases in the Zhejiang province were detected signaling a spread throughout Southern China (Yuezhong et al., 2009). Other Asian countries where the RPW is present include: India, Pakistan, Sri Lanka, Myanmar, Indonesia, the Philippines, and the Gulf states (Abe et al., 2009). In the Western Hemisphere, the RPW was detected in the Caribbean islands of Aruba and Curaçao in 2009 which meant a risk of the weevil being exported to South America (Roda et al., 2011). On October 2010, the USDA's Animal and Plant Health Inspection Service (APHIS) announced the arrival of the closely related species *Rhynchophorus vulneratus* to the city of Laguna Beach, California. Today, the RPW is present in over 70 countries (CABI, 2017).

In Catalonia, Spain, there are over 800 gardens, nurseries and plant traders combined, 24% of them trading with palm trees. Catalonia has five administrative regions, all of them involved in trading palms which increases the risk of RPW expansion (Table 1). Places with large volumes of trade and transport and adjacent to high human population densities likely benefit from prioritizing invasions (McGeoch et al., 2016). This is the reason for selecting Catalonia as our case study.

Table 1 shows that the region with most nurseries trading with palm trees is Girona, followed by Barcelona, Terres de l'Ebre, Tarragona, and Lleida. However, if we look at the total nurseries trading with palm trees as a ratio of all nurseries, Girona has the highest ratio again, but the order changes thereafter with Barcelona being the second to last in terms of this proportion. The fact that palm trees are well present in all the regions and that the RPW can fly over certain distance, are two factors that contributed to the pest spreading quickly from southern to northern Catalonia.

Concerning the flying capacity of RPW, if the food is plenty the gregarious weevils tend to stay in the infested palm until it collapses. Then they may move to a nearby palm if available or fly away searching for new host plants. A mark–recapture study in date palm plantations in the United Arab Emirates indicated that R. ferrugineus can fly ca. 1–7 km in 3–5 days to reach pheromone traps (Abbas et al., 2006; Hoddle et al. (2015)), using computerized flight mills in lab conditions, concluded that ca. 30% of the weevils tested exhibited short-distance flights covering distances similar to those reported by Abbas et al. (2006), and that in the field perhaps ca. 70% may have the capacity to fly long distances (> 10 km) in a relatively short time (24 h).

3. Method

3.1. Cost-benefit analysis

We perform a cost-benefit analysis assessing a set of conservation policies put into place to protect two susceptible species from the RPW: canary and date palms (Generalitat de Catalunya, 2006). The former was reported in 99.69% and the latter in 0.23% of RPW infestation cases in Catalonia (Generalitat de Catalunya, 2011b). Although date palms currently represent a small fraction of reports, we include them because in the absence of canary palms, the RPW would likely shift to that species. The first step of a cost-benefit analysis is to define what constitutes as costs and benefits. We interpret as costs all policy expenditures with the direct aim of preserving and protecting palm trees from the RPW. This covers the prevention and inspection of healthy palms, treatment of infested palms and removal of palms that are too infested to be salvaged, and government expenditure associated with surveying, research, regulation, management and outreach linked to the RPW (Aukema et al., 2011).

To better understand the rationale behind the accounting used in this study, a few remarks on the policies are relevant. From 2005 to 2011, the Catalonian government (GENCAT) was responsible for research and outreach in addition to the management activities (inspection, prevention, treatment and removal costs, excluding replanting). From 2011, onwards, GENCAT continued funding research and outreach but stopped undertaking the management activities described, since their cost had become too large. The latter were left to municipalities and property owners, whose response was uneven. As to private owners, the majority simply gave up and let their palm trees die, without replanting. This was because some owners were not aware of the problem, despite the information campaigns, while others found everything that they had to do to save their palms unattainable. Private owners did not participate much in undertaking the management activities described, either for lack of time, knowledge or money. Most palms infested by RPW's were old and tall trees, planted decades ago, representing the most preferred by the weevils. The weevil's larvae live and feed deep inside the palm crowns (at the top of the trees), so one needs a ladder-crane to reach up there and cut open the crown with a chainsaw to get to the larvae. It is a quite perilous and expensive activity, which in most cases requires a professional gardener. Chemical treatment by endotherapy, i.e. by injecting the palm trunk with an insecticide, also requires a professional gardener, which is again expensive while the treatment does not always work. In practice, in most cases only 'rich' and well-informed private owners could afford the costs of saving their decades-old beautiful canary palms. Although replanting is not explicit in the policies, between 2005 to present, the local media has reported some replanting initiatives (Congostrina, 2017; Caro, 2017). We, therefore, include replanting on behalf of municipalities and property owners throughout the entire period.

To define benefits, we will use the total economic value (TEV) framework (Ledoux and Turner, 2002). It classifies the value of an asset into two main categories: use and non-use values. The main difference between these is that use values are related to the use of the asset while non-use values relate to the motivation for conserving the asset for future generations or for its own sake (existence value). Fig. 2 further illustrates this point by listing indirect and direct values derived from palm trees.

Canary and date palms are non-native to the Iberian Peninsula. They were introduced in ancient times appearing in depictions of late Iberian pottery and coinage dating back to the third to first century BC (Al-Khayri et al., 2015). Although canary palms produce edible fruits, they do not meet the standards required for agriculture. In Spain, the land area dedicated to date cultivation is marginal with an average of ca. 600 hectares since 1961 to present (FAOSTAT Database, 2018). In Catalonia, our region of study, date cultivation is virtually non-existent. Palms are planted in urban areas, resorts, public spaces and private gardens mainly near the coast (Al-Khayri et al., 2015). Currently, they are classified as species of "least concern" on the IUCN Red List of Threatened Species (Beech, 2017), meaning they do not qualify as threatened, near threatened or conservation dependent.

Given their almost exclusive ornamental use, we identify two indirect use values of palm trees as benefits: carbon sequestration and enhanced aesthetic value, the latter being the main reason behind palm tree availability in Catalonia. Although in some places palm trees have direct values (i.e. food and bio-fuel), and indirect values (i.e. shading and hosting wildlife species), in the scope of this article, no direct values were identified. Estimating the costs and benefits in the conservation policy scenario and the scenario without such policy will be the core of our cost-benefit analysis.

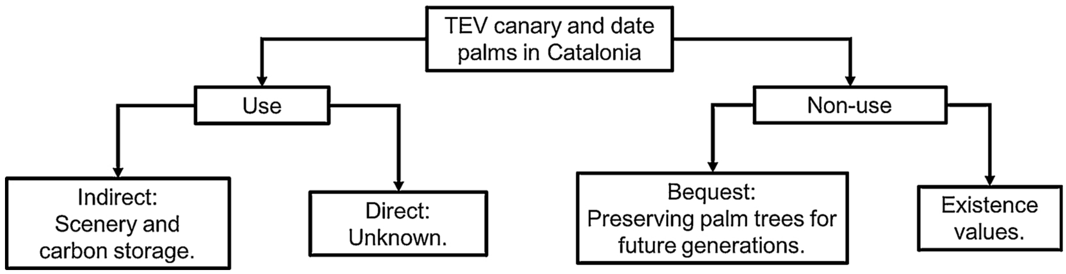


Fig. 2. Total economic value framework for canary and date palms in Catalonia.

3.2. A dynamic model of tree stocks under RPW infestation

We modeled the evolution of tree stocks as a system dynamics problem using stocks, flows and feedback loops since this approach allows us to understand nonlinear behavior over time. The two palm species considered in this study have similar processes with different initialization values so we implemented two separate modules of the same model. From this point onward, susceptible palm species will be referred to as *trees*. The model was designed with three stocks: healthy, infested and treated trees. Fig. 3 illustrates the model. The total amount of trees, S_t, at time t, will be the sum of healthy trees, S_t^H, the sum of infested trees, S_t^I, and the sum of trees in treatment, S_t^T:

$$S_t = S_t^H + S_t^I + S_t^T \tag{1}$$

S_t^H are all susceptible trees with two incoming flows: replanted trees, $F_{t,t-1}^R$, and successfully treated trees, $F_{t,t-1}^{ST}$. Prevention consists of a series of chemical applications to reduce tree vulnerability to the pest. Outgoing flows are trees in which prevention has failed, $F_{t,t-1}^{FP}$. Therefore, the equation for the healthy net flow, $S_t^H - S_{t-1}^H$, between $t-1$ and t is:

$$S_t^H - S_{t-1}^H = F_{t,t-1}^R + F_{t,t-1}^{ST} - F_{t,t-1}^{FP}, \tag{2}$$

Since both successful, $F_{t,t-1}^{ST}$, and failed treatment flows, $F_{t,t-1}^{FT}$, depend on the probability of successful treatment σ

$$F_{t,t-1}^{ST} = S_{t-1}^T \sigma \tag{3}$$

$$F_{t,t-1}^{FT} = S_{t-1}^T (1-\sigma), \tag{4}$$

while the failed prevention outflow, $F_{t,t-1}^{FP}$, - product of the number of healthy trees, S_t^H, and the force of infection at time t, λ_t:

$$F_{t,t-1}^{FP} = S_t^H \lambda_t, \tag{5}$$

Eq. (2) can be rewritten as

$$S_t^H = S_{t-1}^H + F_{t,t-1}^R + \sigma S_{t-1}^T - \lambda_t S_t^H = (1-\lambda_t) S_{t-1}^H + F_{t,t-1}^R + \sigma S_{t-1}^T. \tag{6}$$

The force of infestation, λ_t, is the rate at which susceptible trees become infested per unit time. It is adapted from the force of infection (Vynnycky and White, 2010). The rate of infestation is related to the number of infested trees, S_t^I: the more trees are infested, the more likely more trees will become infested and to the average number of trees that are planted next to each other, denoted by β_t:

$$\lambda_t = \beta_t S_t^I. \tag{7}$$

β_t in turn, depends on the effective contact rate (C_E), defined as the contact rate sufficient to lead to transmission if it occurs between an

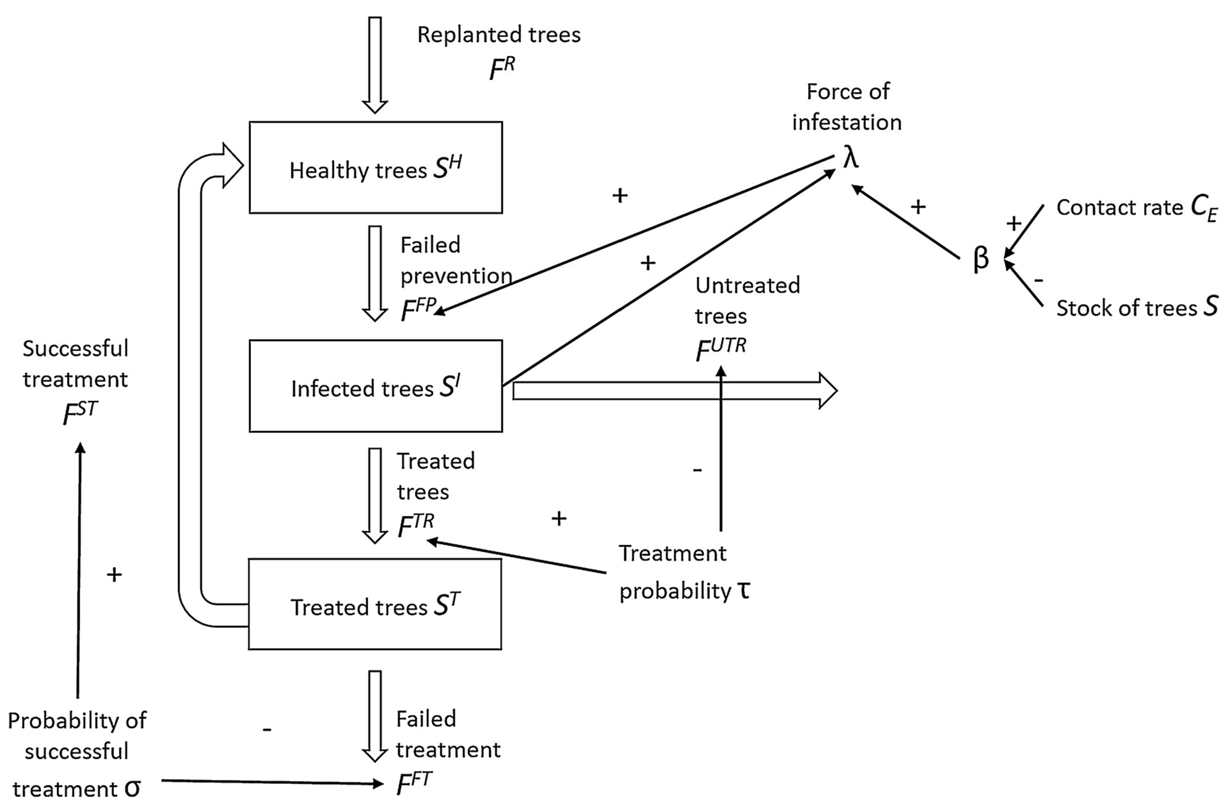


Fig. 3. Evolution of tree stocks in our model.

infested and a susceptible tree (Vynnycky and White, 2010) out of the stock of total population of trees, S_t, at time t:

$$\beta_t = \frac{C_E}{S_t}. \tag{8}$$

Failed prevention, $F^{FP}_{t,t-1}$, is an incoming flow to the infested tree stock, S^I_t. Infested trees are either treated, $F^{TR}_{t,t-1}$, or not treated, $F^{UTR}_{t,t-1}$, leading to two outgoing flows for this stock. Therefore, the equation for the net infested flow between $t-1$ and t is:

$$S^I_t - S^I_{t-1} = F^{FP}_{t,t-1} - F^{UTR}_{t,t-1} - F^{TR}_{t,t-1} \tag{9}$$

Both treated, $F^{TR}_{t,t-1}$, and untreated flows, $F^{UTR}_{t,t-1}$, depend on the probability τ, of being treated:

$$F^{TR}_{t,t-1} = \tau S^I_{t-1} \tag{10}$$

$$F^{UTR}_{t,t-1} = (1-\tau) S^I_{t-1} \tag{11}$$

Based on Eqs. (5), (10), (11) one can transform Eq. (9):

$$S^I_t = S^I_{t-1} + S^H_t \lambda_t - (1-\tau) S^I_{t-1} - \tau S^I_{t-1} = \lambda_t S^H_t. \tag{12}$$

Trees selected for treatment, $F^{TR}_{t,t-1}$, is an incoming flow to the treated trees stock, S^T_t. The latter has outgoing flows representing either failed, $F^{FT}_{t,t-1}$, or successful treatment, $F^{ST}_{t,t-1}$. Thus,

$$S^T_t - S^T_{t-1} = F^{TR}_{t,t-1} - F^{FT}_{t,t-1} - F^{ST}_{t,t-1}. \tag{13}$$

Hence,

$$S^T_t = S^T_{t-1} + \tau S^I_{t-1} - (1-\sigma) S^T_{t-1} - \sigma S^T_{t-1} = \tau S^I_{t-1}. \tag{14}$$

Finally, to keep track of trees that were lost each year due to lack of treatment,[2] $F^{UTR}_{t,t-1}$ or failed treatment, $F^{FT}_{t,t-1}$, we define the lost flow, $F^L_{t,t-1}$, between $t-1$ and t as:

$$F^L_{t,t-1} = F^{UTR}_{t,t-1} + F^{FT}_{t,t-1} = (1-\tau) S^I_{t-1} + (1-\sigma) S^T_{t-1} \tag{15}$$

Similarly, to keep track of trees that have been replanted each year, S^R_t, due to the incoming flow of replanted trees, $F^R_{t,t-1}$, we define the flow of replanted trees $F^R_{t,t-1}$, between $t-1$ and t as:

$$F^R_{t,t-1} = S^R_t - S^R_{t-1} \tag{16}$$

Together, Eqs. (1), (6), (12), and (14)–(16) describe the dynamics as shown in Fig. 3.

3.2.1. Costs

We now introduce equations to calculate costs. These are divided into two main groups. The first is management costs assumed by municipalities and private palm owners from 2011 onwards. These management costs, C_t, for trees in year t are described as the sum of inspection ($C^{inspect}_t$), prevention ($C^{prevent}_t$), treatment costs (C^{treat}_t), replanting ($C^{replant}_t$), and removal costs (C^{remove}_t). The second group is government (GENCAT) costs which includes management costs (except replanting) but also additional activities such as research and outreach. These were provided from 2005 to 2017 and since no specific breakdown was provided by GENCAT so they are added as a lump sum (C^{GENCAT}_t).

$$C_t = C^{inspect}_t + C^{prevent}_t + C^{treat}_t + C^{replant}_t + C^{removal}_t + C^{GENCAT}_t \tag{17}$$

Eq. (17) can be rewritten as

$$C_t = S^H_t (c^{inspect}_t + c^{prevent}_t) + \lambda_t S^H_t c^{treat}_t + F^R_t c^{replant}_t + F^L_t c^{remove}_t + C^{GENCAT}_t \tag{18}$$

with $c^{inspect}$, $c^{prevent}$, c^{treat}, $c^{replant}$ and c^{remove} being the unit costs of inspection, prevention, treatment, replanting and removing trees, respectively.

[2] Here we assume that the mortality rate is 100% among infested trees that were not treated.

3.2.2. Benefits

As mentioned in Introduction, trees provide different benefits (B_t). We will consider two benefits provided by palm trees in Catalonia (Eq. (19)): (1) those derived from carbon sequestration VCS_t, and (2) those derived from forestry adjacency VPA_t, or in this instance, palm tree adjacency.

$$B_t = VCS_t + VPA_t \tag{19}$$

VCS_t, will be calculated using the social cost of carbon, which is an economic indicator that represents the net impacts from global climate change that results from a one tonne increase in carbon dioxide emissions (IAWG, 2016). We can calculate VCS_t as follows:

$$VCS_t = cSCCS_t \tag{20}$$

where, SCC is the social cost of carbon dioxide and c is the amount of carbon sequestered by a tree.

VPA_t can be captured by using the hedonic property-value method. This allows to estimate the aesthetic value trees contribute to property sale prices. Several studies have focused on quantifying forest adjacency (Donovan and Butry, 2010; Sander et al., 2010; Tyrväinen and Miettinen, 2000) with results stating that trees contribute between 0.29 to 5% of property values of private residences. In this article we attempt to capture the value that, specifically, palm trees contribute to property prices. We assume that the total value of a property in Catalonian municipalities, where trees are viable, includes a component of palm trees in addition to other market effects. This component of forestry adjacency is captured in property values (PV):

$$PV = MP + PA \tag{21}$$

where, MP is the market price of a property without adjacent palm trees and PA is the value derived from palm tree adjacency.

3.2.3. Total costs and benefits

The net present value (NPV) of the current management policy against the RPW in Catalonia is the difference between benefits and costs, both transformed into present values as of 2018 (Eq. (22)).

$$NPV_{2018} = \sum_{t=2005}^{T=2017} (B_t - C_t)(1-r)^{T-t} \tag{22}$$

Here, B_t denotes the monetary benefits of the policy, C_t, the management costs and, r, the social discount rate.

To determine the net economic benefit of the policy intervention, we also estimate Eq. (22) for the case without conservation policy ($NPV^{no\ policy}_{2018}$). This is essential to accurately estimate the net contribution of the policy measures as we do not know otherwise how many palms would have been destroyed by RPW and the associated net economic costs.[3] $NPV^{no\ policy}_{2018}$ is calculated running the same model but with parameter settings altered to disactivate replanting, inspection, prevention and treatment activities. While the costs of the "no policy" scenario equal zero, its benefits are determined by the number of palms which would be preserved according to our analysis in case no policy against RPW from the side of Catalan government (GENCAT) or private owners was implemented. Eq. (23) then estimates the net social benefit (NSB) of the policy:

$$NSB_{2018} = NPV_{2018} - NPV^{no\ policy}_{2018} \tag{23}$$

3.3. Estimation of parameter values

As explained earlier, the model has three parameters: effective contact rate (C_E), probability of being treated τ, and probability of successful treatment (σ). Once costs and benefits are computed, they will be evaluated with Eq. (22) which requires a discount rate (r). In

[3] We are grateful to an anonymous referee for pointing this out.

this section, we explain the process to estimate each parameter and selection of the social discount rate used to evaluate the policy.

3.3.1. Effective contact rate (C_E)

Effective contact rate, C_E, is defined as the sufficient contact rate to lead to transmission if it occurs between an infested and a susceptible tree (Vynnycky and White, 2010). Since palm trees in Catalonia are often planted in a single row, a single tree can have two neighboring trees. C_E can take up values $[1, \infty]$ but we limit it to the range [1,2] since a single infested tree can infest up to two trees. As mentioned earlier, the weevils (larvae and adults) tend to stay in the infested palm until it collapses. This generally takes one year although it depends on the level of infestation and the size of the palm crown. Then they may move to a nearby palm if available or fly away searching for new host plants. As far as we know there are no studies revealing what the weevils do exactly when abandoning a collapsing first-infested palm. In principle we assume weevils would prefer moving to the closest uninfested palm trees from the originally infested one, although as mentioned above some may fly away several km colonizing new areas. This assumption is consistent with the fact that these weevils are not excellent fliers.

3.3.2. Probability of being treated (τ)

To elicit treatment efforts and success probabilities, we conducted interviews with three government officials and two local independent firms offering pest management services. The latter included inspection of trees, prevention, and treatment or removal of infested trees. The probability that an infested tree is treated, τ, can take values between [0,1]. No official data exists describing the behavior of private owners while municipality reports are undecided. We include the value 1 as it is consistent with legislation stating that treatment of affected trees is mandatory. However, according to interviewees, there are many instances where infested tree treatment does not occur, which may happen deliberately or accidentally. To capture this uncertainty, we set the lower boundary of τ equal to 0.5 (i.e.$\sim$ coin toss). As a result, we limit our analysis to the interval [0.5,1].

3.3.3. Probability of successful treatment (σ)

Sigma (σ) is the probability that treatment is successful. Again, as it is a probability, it can take values between [0,1]. Success is defined as a tree being sufficiently recovered to rejoin the healthy stock. Interviewees were consistent in their responses stating that once a tree becomes infested, the survival probability is slim to none. For this reason, we assigned [0,0.5] to the interval as it reflects the pessimistic outlook on trees under treatment: from a zero recovery rate up to 0.5 (i.e.$\sim$ coin toss).

3.3.4. Social discount rate (r)

The social discount rate (SDR) is an important parameter in the valuation of public policies. It is defined as the discount rate used by society to give relative weight to social consumption or income accruing at different points of time (Price, 1988). The lowest SDR found for Spain was (3%) (CATSALUT, 2014), used for healthcare project evaluation in Catalonia and the highest was (6%) (Zhuang et al., 2007) used for transport related projects. We present our results with lower and upper bounds. The lower bound will be calculated with the 3% SDR and the highest with 6%.

4. Results

In this section we present calculations of costs and benefits under the conservation policy scenario and for the scenario without such policy. Recall that we consider two types of benefits provided by palm trees (Eq. (19)), those derived from carbon sequestration and others derived from palm tree adjacency. Here we present the results for both calculations. We implemented the model using the deSolve package in R Software (Soetaert et al., 2010). To run the model, two initial stock values had to be defined for $t = 2005$: healthy, S^H_{2005}, and infested, S^I_{2005}, stock of trees. In this section we also explain how these two values were set.

4.1. Healthy tree stock in 2005

Since public tree inventory is decentralized and private owners have the ability to change their landscape, public and private tree records are unavailable. Instead, we estimated tree stocks. To approximate the total stock in 2005 (Eq. (24)), S_{2005}, we first estimate values for 2017, S_{2017}, and add the minimum loss of stock during the invasion period, $S^L_{2017-2005}$:

$$S_{2005} = S_{2017} + S^L_{2017-2005} \tag{24}$$

The minimum loss was provided by the regional government (GENCAT). The process for estimating stock in 2005 will be now explained with more detail.

As previously mentioned, the availability of both palm trees in the region of study is human-mediated given its almost exclusive ornamental use. Due to this type of usage, we identified multiple human-centred factors as explanatory variables (population, GDP per sector and municipality area). Furthermore, we consulted two experts in forestry and ornamental plants who suggested distance to the Western Mediterranean Basin plays a key role in the presence of palm trees. Data for aforementioned variables were retrieved from the Catalonian statistical department (IDESCAT) for all municipalities. Distance was calculated using polygon centroids. GDP per sector was not available for all municipalities, however, it was available for all *comarcas*, which are a higher administrative level. *Comarcas*' GDP per sector was used for municipalities lacking these data.

To collect data on both species, we counted tree occurrences using static maps from Google Maps. To select the sample, we chose a stratified sampling strategy where *comarcas* were defined as strata, since we wanted all *comarcas* to be represented within the sample. Initially, we set out to collect a sample size of 50 municipalities. The total number of Catalan municipalities is 947 contained in 42 *comarcas*. For each strata, we calculated the sample size as: $\frac{Sample \quad Size}{Total \quad population} * Strata \quad size$. For example, for the comarca Baix Camp with 28 municipalities, calculations were: $\frac{50}{947} * 28 = 1.48 \approx 2 \quad municipalities$. Repeating this process for all comarcas and rounding decimals upwards yielded 73 municipalities. Once the sample was selected, the following procedure was done for each of the 73 municipalities. First, using a 'shapefile' for the region and the R Packages sf and GGMap (Kahle and Wickham, 2013), a grid for each municipality was produced where each cell in the grid represented a separate image (Fig. 4). After this image-generation process was done for each municipality, each of the 25,000 images was individually assessed for palm trees of each species.

Once observations were collected, we proceeded to decide what model to use to estimate the distribution of palm trees. The Bayes Information Criterion (BIC)[4] was used as the criterion to select the final model. This was implemented with the *dredge* function of the MuMln package for R software. This function generates a set of models with combinations of fixed effect terms in the global model. The model was specified as follows for each municipality: percentage of GDP derived from agriculture, services, construction, and industry; distance to the coastline, population, and area. With these seven parameters we were able to test ($2^7 = 128$) models to see which one minimized the BIC.

In terms of significant estimators this meant, distance to the

[4] Selecting a model based on an information criterion is standard in econometrics, see Winker (2001) and Savin and Winker (2012). Its idea is based on finding a parsimonious model with high explanatory power. $BIC = -2\log(likelihood) + \log(n)\#parameters$, where n is the sample size. We thus have a tradeoff between the goodness of fit $\log(likelihood)$ and the model complexity represented with $\log(n)\#parameters$. The latter helps us to avoid overfitting the data. We choose the model that has the smallest BIC.

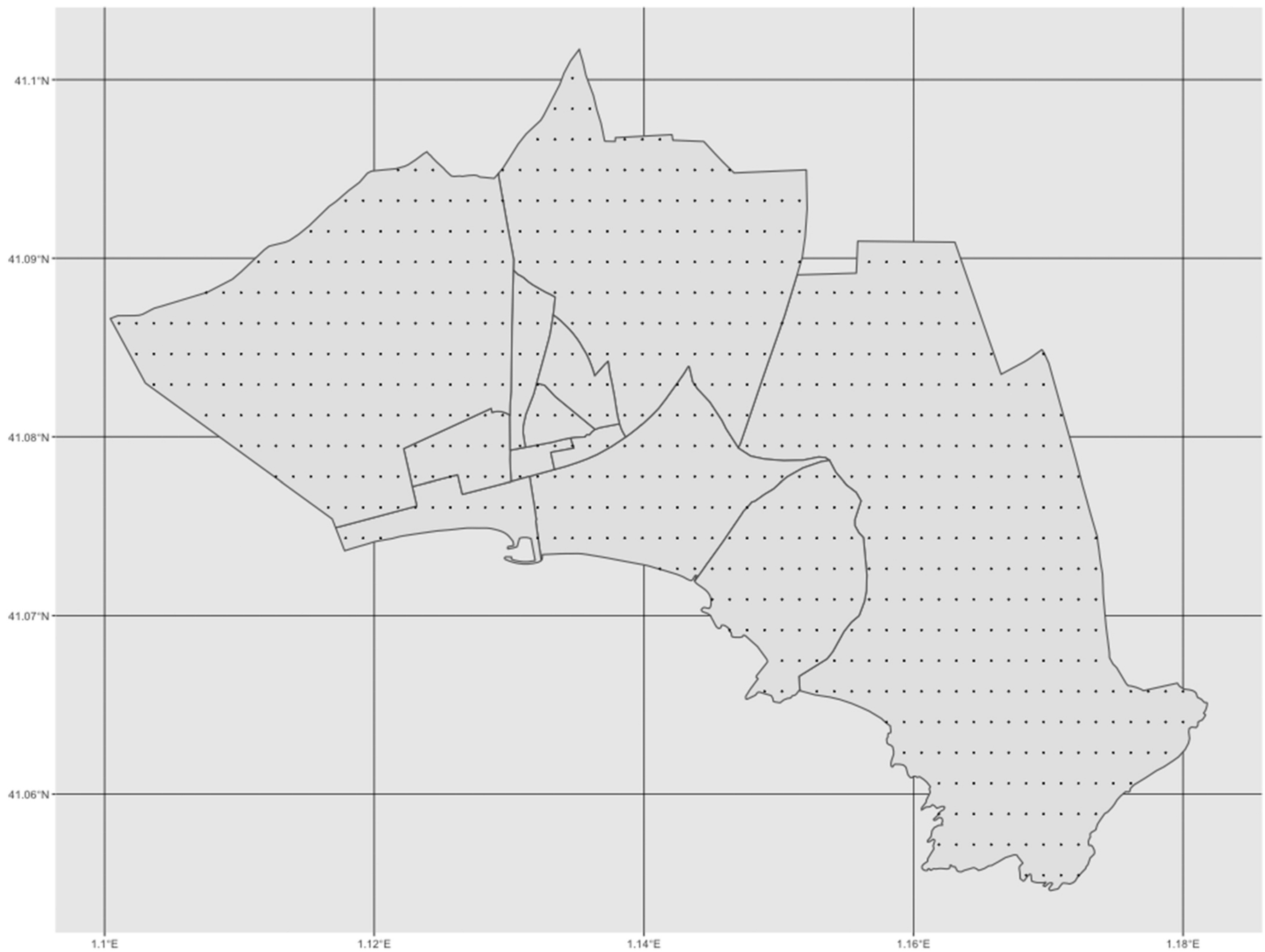


Fig. 4. The image shows a municipality grid for Salou, Catalonia. Each point on the map represents a separate image printed at a resolution sufficient to count palm tree occurrence.

coastline, and industry contribution to GDP (%), for date palms; and distance to the coastline, population, and service contribution to GDP (%), for canary palms. Tables C1 and C2 in Appendix C summarize the model estimates for date and canary palms, respectively. To show other possible models in order of ascending *BIC* we have included the two next best models in line.

Using these models, we estimate the number of healthy date and canary palms in Catalonia. Estimations appear in Fig. 5 as a function of distance to the coast. For most municipalities, canary and date palms are located closest to the coast. Date palms are inversely related with industry contribution to GDP (%) and canary palms have a positive relationship with services contribution to GDP (%). Finally, recall that these estimates correspond to 2017 estimates. To obtain 2005 estimates, we add the reported loss for date (14) and canary palms (8946) provided by GENCAT. Final estimates for date and canary palms were 22,737 and 27,567.

4.2. Infested tree stock in 2005

To initialize our model, we need to estimate the number of infested trees at the beginning of the epidemic (2005). Since no exact information was available, we tried to estimate this initialization value by running several simulations of the model while varying the initial infested stock. As reference, we used reports for infested stock per year. The reports are government (GENCAT) records collected to monitor the evolution of the pest between 2005 and 2014. According to interviews with GENCAT officials, these were collected through non-systematic communications with stakeholders. For example, private owners or municipalities would contact GENCAT and report any noticeable palm tree changes. This is to say that these reports do not accurately reflect the RPW's spatio-temporal infestation pattern, but rather provide a minimum volume of affected palm trees.

If the simulation contained the reported infested stock, then it is more likely the initialization parameters are appropriate. We used Latin Hypercube Sampling (McKay et al., 1979) to sample parameter values from a multidimensional distribution represented in Table 2 and in this way initialize four alternative runs. This was implemented with the R FME package (Soetaert et al., 2010). Four simulations with the random values for each parameter are presented in Fig. 6.

Based on the infested stock for canary palms, we note that simulation four, best fits the reports up to 2008 (red line). Afterwards, there is a decline in reports, including a sharp decline in 2011. We infer this is due to reporting methods and not to a sudden change in the actual patterns of infestation. This inference is supported by reports from the subsequent year (2012), where an increase falls within simulation three. From these observations, we assume appropriate initialization parameters for infested stock are between those of simulations three and four, which correspond to values between [3–5] for date palms and [365–975] for canary palms. These selected initialization values are summarized along with other parameters in Table 2. These will be used for our cost calculations.

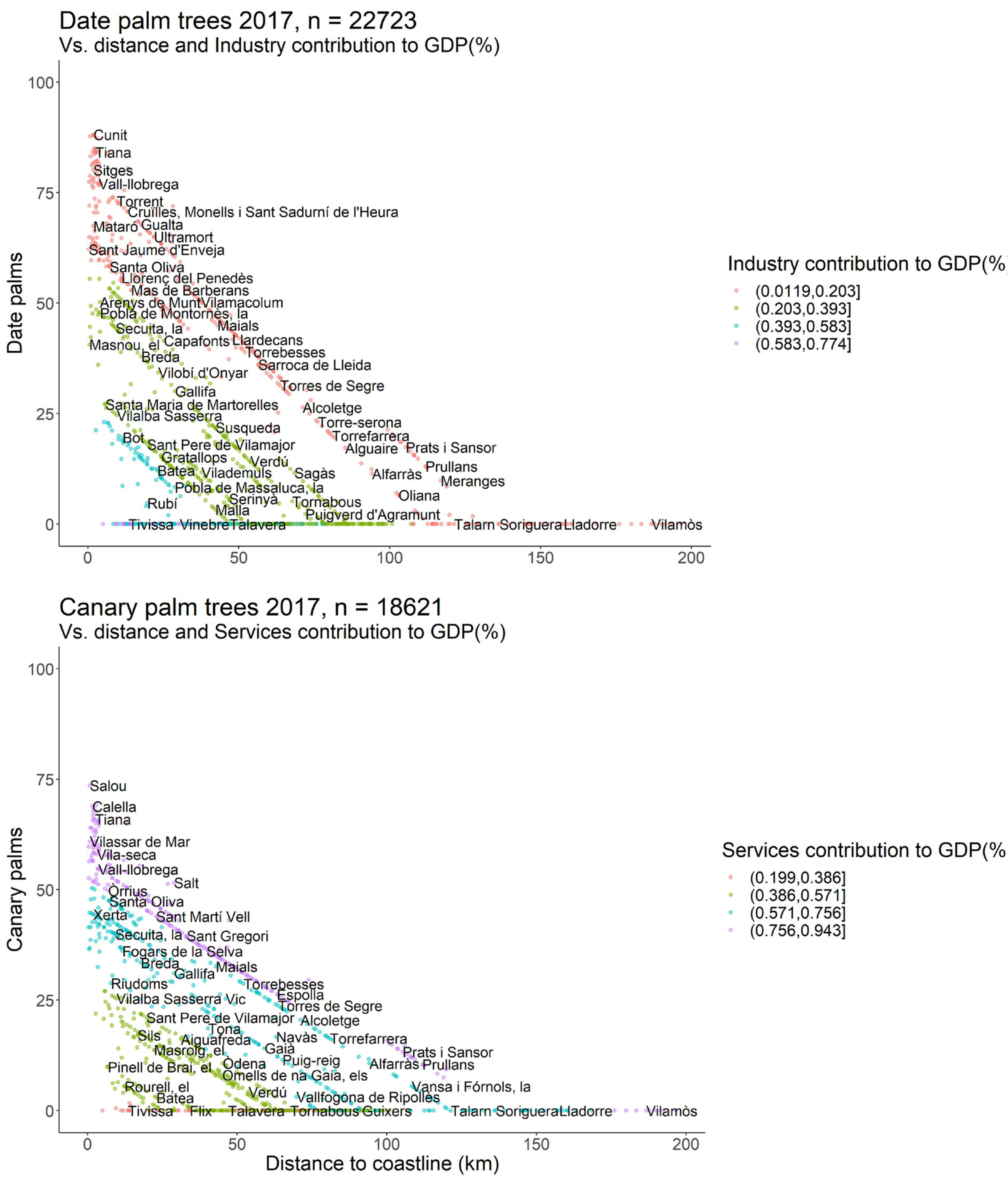


Fig. 5. Healthy palm tree estimations in 2017. Note: The upper plot corresponds to date and the lower one to canary species estimations. Both estimations are plotted against distance to the coastline. Industry contribution to GDP (%) was a significant estimator for date palms and services contribution to GDP (%) for canary palms, these two variables have been included with color. Municipalities with a lower industry contribution to GDP (%) have more date palm trees than those with a higher industry contribution to GDP (%). Similarly, those with a higher services contribution to GDP have more canary palms than those that do not. Overall, there are more date than canary trees. In both cases distance to the coastline has an inverse relationship with palm tree occurrence. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

With the parameters in place to run the model, we can estimate the volume of date and canary trees for each year between 2005 and 2017. Each management activity has a different value and in some cases must be performed more than once each year. For example, replanting and removing a tree can be done only once. Inspection and prevention on healthy trees should be done at least twice per year. Finally, when a tree is infested, it needs to be treated around five times in one year. A summary of these unitary costs along with their frequencies are summarized in Table 3.

4.3. Costs

Costs include government (GENCAT) and management costs paid by

Table 2
Initialization and parameter values for both species.

Values	Date	Canary
Effective contact rate	[1, 2]	[1, 2]
Treatment prob (τ)	[0.5, 1]	[0.5, 1]
Successful treatment prob (σ)	[0.0–0.5]	[0.0–0.5]
Healthy stock 2005	22737	27567
Infested stock 2005	[3–5]	[365–975]

Table 3
Unitary costs for management activities per tree.

Type	Value (€)	Yearly frequency
$c^{inspect}$	10	2
$c^{prevent}$	20	2
c^{treat}	30	5
$c_t^{replant}$	500	1
$c_t^{removal}$	500	1

municipalities and private palm tree owners (Eq. (17)). Data on GENCAT costs were reported by GENCAT as a lump sum and are presented as such. Fig. B1 (Appendix B) illustrates GENCAT costs brought to present value. Costs start rising from 2005 onwards, peak in 2009 and decline afterwards. This is due to the transfer of costs to municipalities and private owners in 2011. Before 2011, research, outreach and management costs (with the exception of replanting) were contained in GENCAT costs, these are summarized in Appendix B (Table B1) and brought to present value. After 2011, private owners and municipalities assume management costs and continue paying for replanting.

We now report costs per species and management category brought to present value. We include a lower and upper bound derived from Table 2. Eq. (17) was applied to the lower and upper bound tree estimates. Tables B2B5 in Appendix B summarize each activity along with total management costs. Table captions summarize the all year total. To illustrate each management category, Figs. B2 and B3 (Appendix B) present this cost breakdown. For date palms, the costliest management activities were inspection and prevention, which makes sense since infestations for this species have been less reported. For canary palms, removal of infested trees has been the most expensive activity. This is consistent with the decline of trees and the high removal cost per tree. Table B6 (Appendix B) summarizes lower and upper bounds for each species and total costs (Fig. B4 in Appendix B). The lower bound has a less steep decline than the upper bound, as canary costs are higher than date costs for all years and it seems that from 2017 onwards, date costs overtake canary costs. In the upper bound where canary decline is stronger, costs are initially high but due to a reduction in overall stock, there is a decline of treatment and removal costs.

4.4. Carbon sequestration benefits

In order to estimate benefits associated to carbon sequestration, we provide an interval per tonne for the social cost of carbon (*SCC*), to reflect uncertainty about its value. The minimum value for the range was 36 USD, as it was lowest *SCC* found (EPA, 2013). Similarly, the maximum found and used was 125 USD (van den Bergh and Botzen,

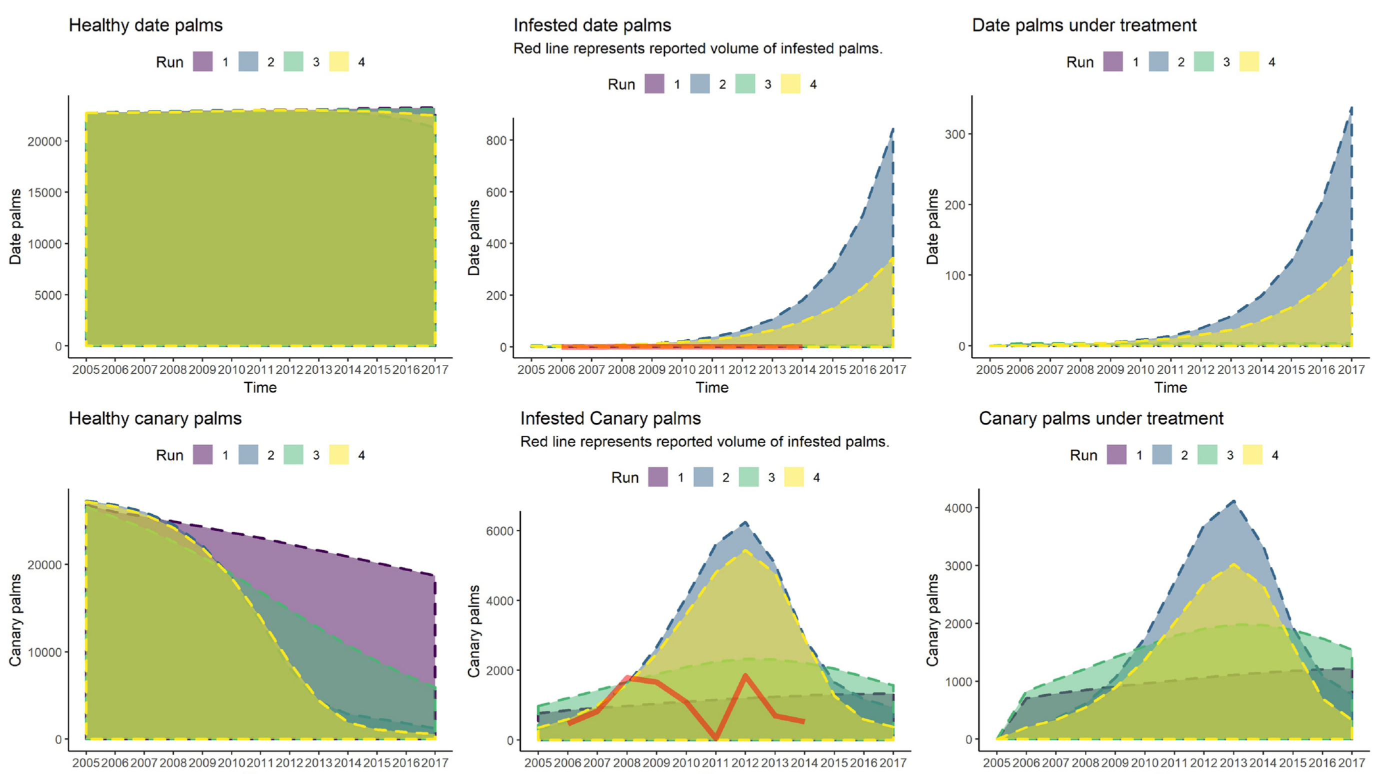


Fig. 6. Four simulations of healthy, infested and treated stocks with different parameters and initialization values. The upper plots correspond to date palms while the lower ones to canary palms. The parameter value have been drawn from the parameter ranges stated in Table 2 using Latin Hypercube Sampling. The main difference between the two species is that date species decline is less severe than canary decline. This difference is based on the preference the RPW has for it. The red line appearing in the infested palms stock corresponds to reports provided by GENCAT and collected over the course of the epidemic from municipalities and private palm tree owners. From the simulations, it appears simulations three (green) and four (yellow), are the best fit as they contain reports (red line). (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

Table 4
Social cost of carbon dioxide per tonne. Present values (PV) correspond to transformed values from published date up to 2017. PV low h uses the lower (higher) bound discount rate.

Source	USD	Published	EUR	PV low	PV high
EPA (2013)	36	2007	31.7	51.3	72.4
van den Bergh and Botzen (2014)	125	2014	110.0	144.9	167.3

2014). These values were converted from US dollars to euros at an exchange rate of 0.88. Table 4 summarizes original values, publication date, currency conversion, and present values (PV) transformed from their publication date to 2017.[5] There are two PV estimates because of the lower and upper bound discount rates being used throughout the article.

In our total benefits calculation, we will present ranges using both the lowest (USD 36) and highest (USD 125) *SCC*. The lowest (highest) bound for each species will be combined with the lowest (highest) *SCC*. By applying Eq. (20) using a value of 0.191 tonnes per year for 1000 trees (Aguaron andMcPherson, 2012; Chaparro and Terradas, 2009) we obtain Table B7, which summarizes the results from this calculation. In particular, columns two to seven provide estimations of carbon sequestration benefits due to trees being preserved under the policy scenario. Since carbon sequestration values are directly proportional to tree stock, lower and upper bound carbon sequestration estimates vary according to estimation of this stock. The upper estimates show a stronger decline over time, explaining the marked difference between 2005 and 2017 values. In columns eight to thirteen in Table B7 we then present the carbon sequestration benefits, net of deducting the number of trees which would have been preserved under no policy scenario. the results show that the population of date palms is hardly affected by the policy, while the largest benefit is associated with the higher estimate of the canary palms, where approx. 9000–12, 000 trees are preserved by the end of 2017. Overall the carbon sequestration benefits are, however, very small, ranging from 604 to 1797 euros.

4.5. Property value benefits

We now present estimations for benefits derived from palm tree adjacency. To calculate these we used data provided by Idealista S.A., an online marketplace for buying and renting properties in Spain, Portugal, and Italy. The response variable used for the regression was the asking or listed price for a property. As actual sale prices for properties in Catalonia could not be obtained, we used asking price as a proxy.

The dataset contained 2118 observations describing detached properties on sale in Catalonia during October 2018. The properties were distributed evenly amongst Catalonia with at least 50 observations per *comarca*, which we defined earlier as our strata. Upon initial exploration of the dataset, there were 8 numerical and 21 character variables. The first step was to identify categorical variables within all the string variables. Here, some were identified as categorical and others related to the unique property address were left in string format. The second step was to deal with missing values in the data. Most of these relate to variables that signaled the presence or absence of a feature (i.e. ~ swimming pool, wardrobe, terrace, garden, etc.). Since these missing values were missing presumably because the property missed a feature, and thus at random, we could have removed the feature. However, we wanted to keep as many observations as possible so we performed imputation.

Since we were interested in capturing the aesthetic value derived from palm-tree adjacency, two variables were created to count the occurrence of each species on properties or within their immediate surroundings.[6] Using the GGMap package for R software (Kahle and Wickham, 2013) and each unique property address, individual images were printed. Each image was screened for either date or canary palms. After removing duplicate properties and addresses which were not found on Google, the dataset had 1591 observations. Each variable was plotted to look at the distributions. Outliers were removed for several variables since there cases listed as independent homes that likely were not part of the same sample (i.e. independent houses with more than 12 floors). In the case of the response variable, the median price per home was €229,000 and the 95th quantile was €750,000 but the maximum sales price was €2,575,000, which skewed the distribution. A similar procedure was followed for all variables that were not boolean. After removing outliers, the dataset had 1169 observations and 9 variables including the response variable. This means we had 8 regressors.

To determine whether date or canary palms contribute to aesthetic values, we fitted a global model with all regressors. The t-tests for date and canary palms did not produce significant p-values. To select the model that best estimated the price variable, we again selected the BIC to compare models. We tested all possible models ($2^8 = 256$) aiming to minimize the BIC. The model with the lowest BIC included number of bathrooms, constructed area, constructed year, garden, swimming pool, wardrobe, and plot size as significant regressors (see Table C3 in Appendix C). Overall, we were not able to capture any values associated with palm tree occurrence.

The hedonic price method applied to the housing market does include aesthetic benefits, as these are one determinant of the house prices. Given that the cost of a new palm tree is between €500 and €1500 – depending on the size of the tree and type of palm species (canary or date) – this is a relatively small cost compared to an average house with garden (several hundreds of thousands of €or more), implying a value share of considerably less than 1%. This might explain why no significant effect was found. To make a robustness check on how large the ornamental value of palms could be, we estimate the lower and upper bound of the value of palms preserved due to the conservation policy. In particular, knowing that with a policy 31,993 to 34,998 palms are preserved in 2017 while under no policy scenario this range would have been 20,305–24,223. By applying the estimated expenditure €500–1500 for planting new palms, as mentioned earlier, we arrive at an estimate of the aesthetic value of palms preserved by the conservation policy, namely €5.85–14.95 million.

4.6. Total costs and benefits

We obtain Fig. 7 by merging total costs and benefits for each bound (lower and upper) in a single plot. The costs include government (GENCAT) and management costs paid by municipalities and private owners from 2011 onwards. By applying Eq. (23) between 2005 and 2017, the net social benefits (NSB) for the lower bound were €-34,137,730. For the upper bound, the NSB is €-43,727,042. The benefits associated with carbon sequestration are much lower than total costs. If, in addition, we add the ornamental value of palms as estimated in Section 4.5, the costs would still dominate the benefits: €-34,137,730 + 5,844,000 = -28,293,730 as a lower bound and €-43,727,042 + 14,947,500 = -28,779,542 as an upper bound.

5. Discussion

The costs of pest management, eradication and replacement of RPW infested palms were previously estimated for Italy, Spain and France by

[5] Here we assume that the social cost of carbon rises at the discount rate as motivated by van der Ploeg and Withagen (2015).

[6] As immediate surrounding we consider the whole area surrounding the property on the Google maps satellite images of approximately 100 by 100 m size.

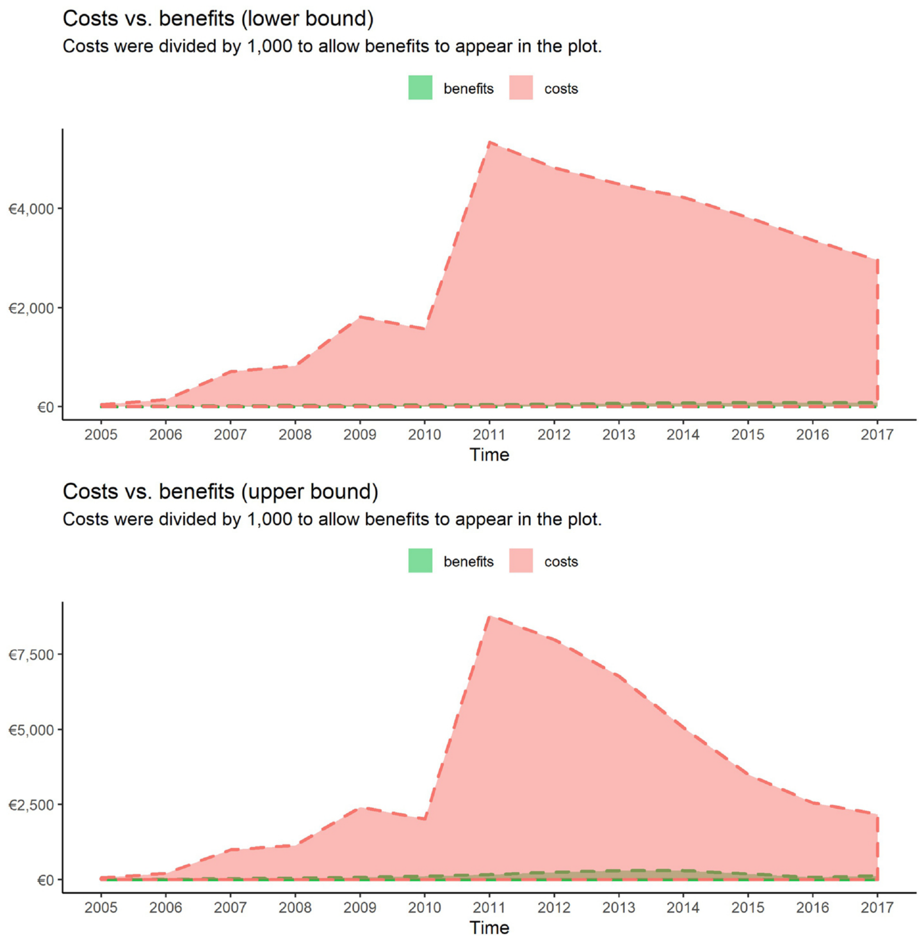


Fig. 7. Total costs vs. benefits for the RPW policy between 2006 and 2017. Costs include GENCAT and management costs for date and canary trees. Benefits include carbon sequestration. In both instances, benefits derived from palm trees are dwarfed in comparison to the expenditure to preserve them.

the FAO. Up to 2013, the combined costs for all three countries were reported at €90 million with an increase of €110 million by 2023 (FAO, 2017). Our cost calculations concur with the magnitude of FAO estimates. We find the policy costs between 2005 and 2017 ranged between €34.1 and €43.7 millions. Those estimates include government expenditure and management activities for both palm species. In contrast, benefit calculations ranged from €604 to €1797 if only carbon sequestration benefits are included. If we take, in addition, the ornamental value of palms into account, the benefits would increase to €5.85–14.95 millions. Because the costs far outweighed the benefits, we did not find the conservation policy justified based on the analysis presented in this article.

Countries where a similar cost-benefit analysis could yield results that support a similar policy are, for example, Tunisia and Egypt, where production of date fruits is not insignificant and the RPW is allegedly a major threat (Speakman Cordall, 2017). Another place where this type of conservation policy could be justified is the Canary Islands since canary palms are a native species, unlike in the Iberian Peninsula. When a species is native, conservation efforts from a socio-cultural perspective are more likely to be justified. In these islands, the RPW was reported in 2007 and declared completely eradicated by 2016. Eradication efforts included the inspection of 706,081 palms, of which 209,537 were treated, 659 removed and destroyed, and 681 weevils were caught (Gobierno de Canarias, 2016).

The larger mandate that underpins the conservation policy here analyzed is 2007/365/CE (see Appendix A). This policy was released for EU member states with susceptible trees stating mandatory measures, including the ones here described as management activities. Within EU publication related to this policy, we could not find an invasion analysis included justifying the EU wide mandate. Prioritization of budgets for biological invasions should be based on analysis that include species, pathways and sites, and impacts (McGeoch et al., 2016).

Understandably, an argument against the type of cost-benefit analysis here performed is that social and cultural values are hard to capture and quantify. Non-monetary values are important, as acknowledged by the TEV framework. In a world with unlimited resources, conservation of natural assets without any direct or indirect benefits, could be possible. In practice, when dealing with biological insect invasions that affect human health, livelihoods, and threatened species, prioritizing policies with the best cost-benefit ratio is critical.

6. Conclusions

In the context of rising biological invasions with costly impacts, it is important to develop a rational conservation policy. In this article we assess a conservation policy implemented in the context of the arrival of an invasive species to the Iberian Peninsula, which eventually spread to

Mediterranean coastal areas in the EU. To do so, we select a policy published in response to the arrival of an invasive insect to the Iberian Peninsula in the early nineties - the red palm weevil (RPW), a beetle that affects susceptible palm tree varieties. To assess the policy, we perform a cost-benefit analysis within the Autonomous Community of Catalonia, Spain. This is done in several steps. First, we interviewed stakeholders from government and industry to properly understand all associated costs. Second, we estimated the number of palm trees in the region at the beginning of the infestation and dynamically modeled their decline until present. Third, based on the Total Economic Value (TEV) framework we identify two quantifiable benefits derived from palm tree existence: carbon sequestration and enhanced aesthetic value derived from palm-tree adjacency and captured through property prices. Fourth, we calculate the costs and benefits associated with the conservation policy and the scenario without such policy. Finally, we compute the net social benefits of the policy. We find that the costs of the policy far outweigh the benefits. This is likely due to the almost exclusive ornamental use of palm trees in Catalonia, Spain.

Acknowledgments

Servei de Sanitat Vegetal de la Generalitat de Catalunya provided interviews, records and further information on the pest in Catalonia. Dr Martí Boada and Dr Jaume Marlès kindly gave feedback on the estimation of the number of palm trees. Joan Freixa and Josep Muntaner shared insights on private palm tree owners and their behavior. Cristian E. Nuno from the University of Chicago Urban Labs provided guidance on calculating distances to the coastline. Idealista S.A. shared the dataset to estimate palm tree adjacency benefits.

Funding

This research was funded by the Spanish Ministry of Science, Innovation and Universities, through the “María de Maeztu” program for Units of Excellence (MDM-2015-0552). Ivan Savin acknowledges financial support from the Russian Science Foundation [RSF grant number 19-18-00262].

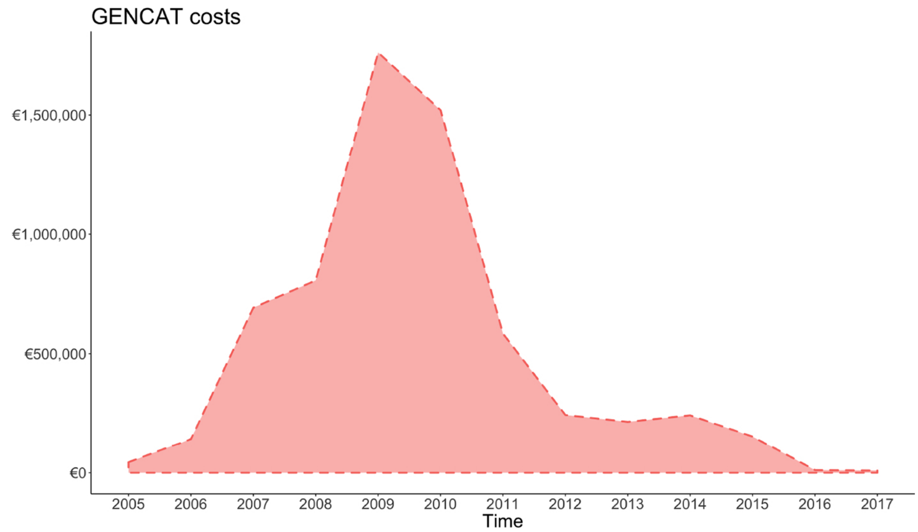
Appendix A. Summaries of the policies selected and related legislation

This section expands on details related to the policy assessed in this article. It summarizes all GENCAT orders and includes the EU legislation that underpins local RPW rules.

- Order ARP/343/2006: On July 3rd 2006, RPW presence in Catalonia is confirmed. Pest prevention and management is declared of public interest. Costing implications based on articles 8, 9 and 12 include inspection, detection, destruction and indemnification to property owners. The total costs for these activities have been provided by the regional government, which also include survey, research, regulation, management, and outreach costs (Generalitat de Catalunya, 2006).
- 2007/365/EC: On 25th May 2007, the EU Commission publishes a decision based on the RPW spread along the Iberian Peninsula. The decision specifies palm tree vulnerability and announces a series of measures to be taken by member countries, including delimitation of areas at RPW risk (EU Comission, 2007).
- 2008/776/EC: On 6th October 2008, the previous decision 2007/365/CE is modified by expanding the range of vulnerable palm trees (EU Comission, 2008).
- Order AAR/226/2009: GENCAT updates Order ARP/343/2006 by redefining and specifying actions to be taken, namely delimitation, inspections, pruning, treatment, and destruction of affected palm trees. This update is based on a decision made by the EU Commission. All costs continue to be borne by the GENCAT (Generalitat de Catalunya, 2009).
- 2010/467/EC: On 28th August 2010, the 17th August 2010 decision is corrected. The added statement is that if routine inspections from the past three years show that eradication does not seem feasible within a year, local measures should focus on pest contention and suppression within the selected area but maintaining eradication as a long-term objective (EU Comission, 2011).
- Order AAR/2802/2010 and Resolution AAR/2802/2010: On 20th October 2010, the delimited area containing the pest is published at municipality level (Generalitat de Catalunya, 2010).
- Order AAM/56/2011: On April 12th 2011, the Order ARP/343/2006 is modified. The main changes are found in articles 8, 9, 11, 12. The key change is that from this year onwards, GENCAT continues to inspect each year, publish the delimited area, and inform local authorities through outreach activities. Treatment and removal costs, however, are transferred to municipalities and private owners. The transferred activities are mandatory and their procedure is carefully described including the type of chemicals to use, the times of the year when they should be done, and the specific places where trees should be destroyed (Generalitat de Catalunya, 2011a).
- Repeal to Decision 2007/365/EC: On 21st March 2018, the EU Commission decides to repeal the 2007/365/EC decision to prevent the introduction and spread of the RPW within the European Union. The repeal started applying on 1st October 2018. The decision states annual surveys by countries enforcing the decision reported that the pest had spread to all vulnerable areas (EU Comission, 2018).

Appendix B. Detailed results

This appendix expands on the results by providing detailed costs and benefits. The tables and figures that appear in the main text are aggregated versions of the ones that appear in this section.



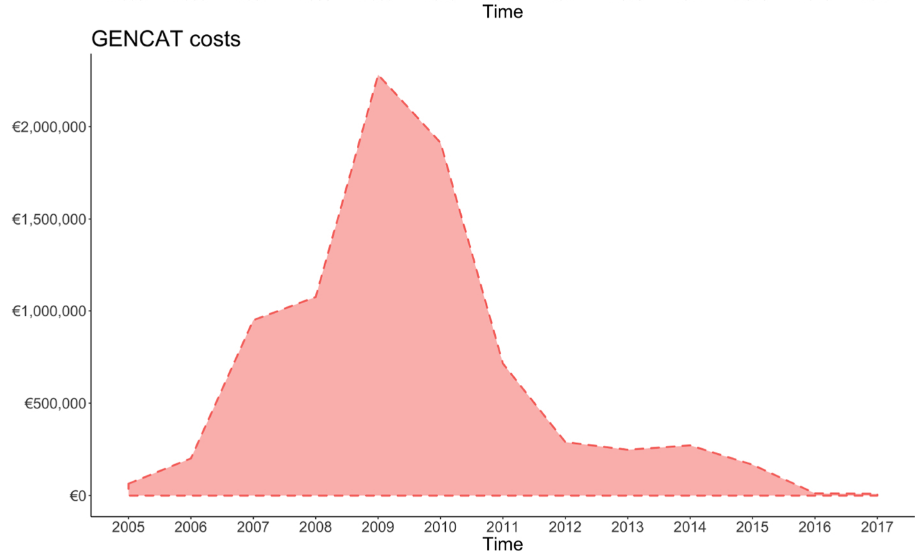


Fig. B1. Government (GENCAT) expenditure over time on the palm tree conservation policy associated to the red palm weevil. Costs increase each year between 2005 and 2009. In 2011, expenditure is more than halved. This is due to inspection, prevention, treatment, and removal costs being transferred to municipalities and private tree owners.

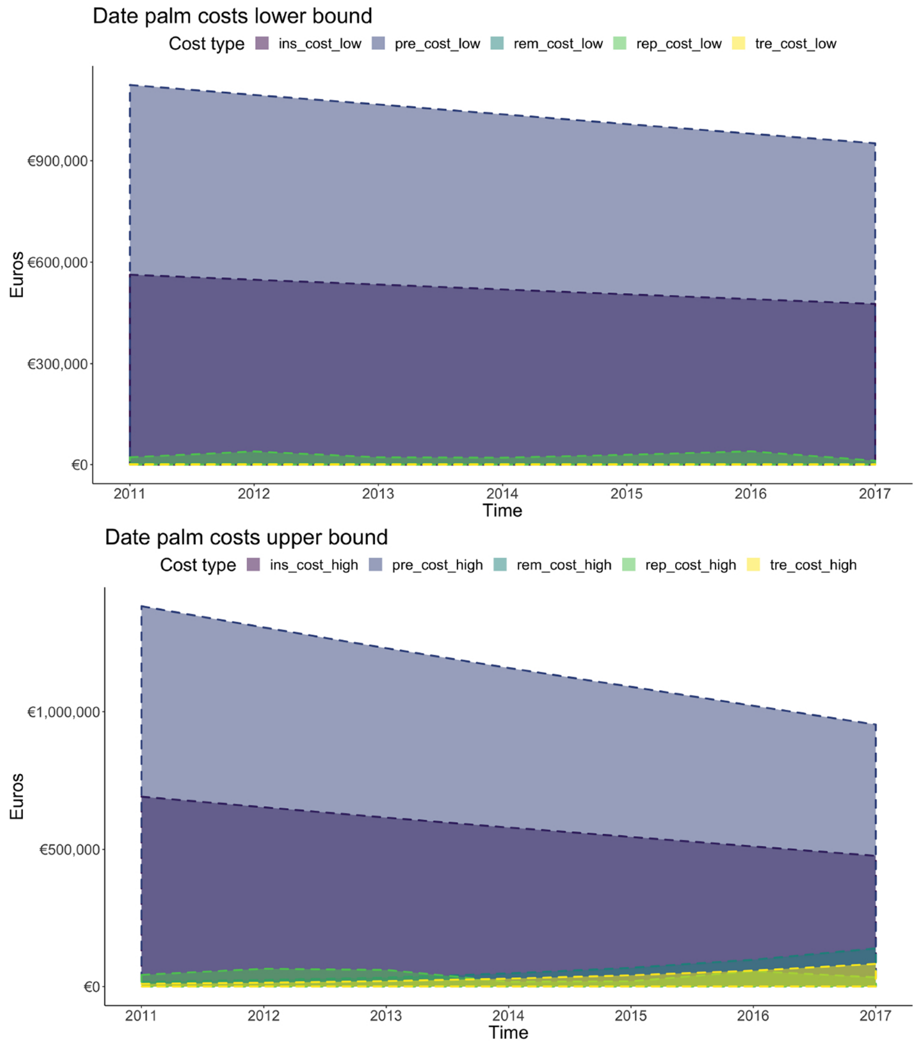


Fig. B2. Management costs for date palms by activity for lower and upper estimations between 2011 and 2017. The lower and upper bounds show a similar pattern for prevention and inspection costs. The main difference between the two bounds is that removal costs are increasing for the upper bound. This is due to a larger volume of date palms and the significant (in comparison to the others) unitary costs of removing an infested tree.

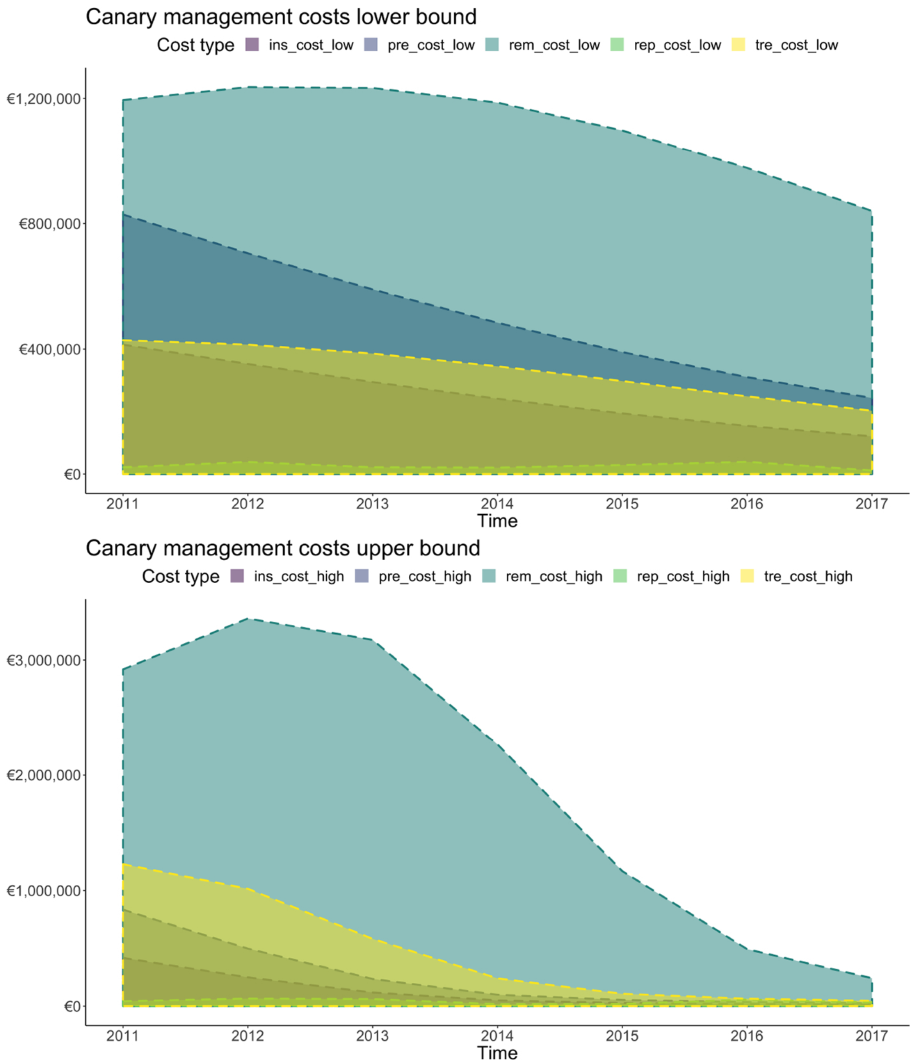


Fig. B3. Management costs over time for canary palms by activity between 2011 and 2017. The biggest expenditure is on removal costs since this species experienced the sharpest decline. In the upper bound, as the epidemic unfolds, more trees become infested that require large volumes of treatment. In the lower bound, the second largest cost is prevention.

Fig. B4. Total costs over time for GENCAT, date and canary palms between 2005 and 2017. GENCAT costs are the same for both bounds. In the upper bound (bottom), GENCAT costs appear smaller in contrast to canary palms. A bigger estimation of palms, led to a larger expenditure on removal costs. In both scenarios, the second largest cost is associated to date palms.

Table B1
GENCAT costs associated to RPW expenditure over time.
Source: GENCAT (2017).

		Present value	
Year	GENCAT	Low	High
2005	30,000	45,378	67,827
2006	100,000	146,853	213,293
2007	500,000	712,880	1,006,098
2008	600,000	830,540	1,138,979
2009	1,350,000	1,814,287	2,417,644
2010	1,200,000	1,565,728	2,027,375
2011	475,000	601,716	757,078
2012	203,820	250,673	306,470
2013	185,232	221,177	262,755
2014	215,110	249,371	287,866
2015	140,000	157,571	176,747
2016	10,545	11,523	12,559
2017	8857	9397	9952

Note: Total cost of the evaluated policies for all years ranges from 6,617,094 to 8,684,643.

Table B2
Management costs (lower bound) for date palms over time.

Year	Replanting	Inspection	Prevention	Treatment	Removal	Total
2011	22,168	579,801	1,159,601	950	3167	1,765,687
2012	39,971	564,586	1,129,172	922	3075	1,737,726
2013	22,090	550,052	1,100,104	896	2985	1,676,127
2014	20,867	535,075	1,070,149	869	2898	1,629,858
2015	29,826	520,255	1,040,510	844	2814	1,594,249
2016	40,431	505,539	1,011,078	820	2732	1,560,600
2017	11,670	490,942	981,884	796	2652	1,487,944

Note: Total cost of all activities for all years is 11,452,192.

Table B3
Management costs (upper bound) for date palms over time.

Year	Replanting	Inspection	Prevention	Treatment	Removal	Total
2011	44,628	733,043	1,466,085	10,280	18,329	2,272,365
2012	68,415	692,151	1,384,302	14,435	24,810	2,184,113
2013	63,833	652,263	1,304,527	20,852	34,044	2,075,520
2014	18,066	614,219	1,228,438	30,110	50,183	1,941,016
2015	20,200	577,987	1,155,974	42,987	71,961	1,869,110
2016	58,955	541,412	1,082,824	61,278	103,023	1,847,492
2017	34,832	505,193	1,010,386	86,798	146,630	1,783,839

Note: Total cost of all activities for all years is 13,973,455.

Table B4
Management costs (lower bound) for canary palms over time.

Year	Replanting	Inspection	Prevention	Treatment	Removal	Total
2011	22,168	426,623	853,246	441,216	1,230,667	2,973,20
2012	39,971	363,133	726,265	426,705	1,273,534	2,829,608
2013	22,090	303,767	607,534	397,261	1,270,472	2,601,124
2014	20,867	249,105	498,210	355,607	1,222,455	2,346,243
2015	29,826	200,993	401,987	307,095	1,130,574	2,070,475
2016	40,431	159,932	319,863	257,009	1,006,402	1,783,636
2017	11,670	125,823	251,645	209,740	865,164	1,464,042

Note: Total cost of all activities for all years is 16,069,048.

Table B5
Management costs (upper bound) for canary palms over time.

Year	Replanting	Inspection	Prevention	Treatment	Removal	Total
2011	44,628	442,452	884,904	1,301,775	3,094,456	5,768,216
2012	68,415	263,556	527,113	1,076,073	3,561,348	5,96,505
2013	63,833	124,404	248,808	616,843	3,364,727	4,418,616
2014	18,066	51,923	103,846	254,330	2,400,777	2,828,942
2015	20,200	28,077	56,155	112,865	1,237,859	1,455,156
2016	58,955	18,770	37,541	67,709	521,665	704,641
2017	34,832	13,169	26,337	48,708	255,619	378,664

Note: Total cost of all activities for all years is 21,050,741.

Table B6
Total costs for canary and date palms over time.

	Lower				Higher			
Year	GENCAT	Date	Canary	Total	GENCAT	Date	Canary	Total
2005	45,378	0	0	45,378	67,827	0	0	67,827
2006	146,853	0	0	146,853	213,293	0	0	213,293
2007	712,880	0	0	712,880	1,006,098	0	0	1,006,098
2008	830,540	0	0	830,540	1,138,979	0	0	1,138,979
2009	1,814,287	0	0	1,814,287	2,417,644	0	0	2,417,644
2010	1,565,728	0	0	1,565,728	2,027,375	0	0	2,027,375

(*continued on next page*)

Table B6 (*continued*)

	Lower				Higher			
Year	GENCAT	Date	Canary	Total	GENCAT	Date	Canary	Total
2011	601,716	1,765,687	2,973,920	5,341,323	757,078	2,272,365	5,768,216	8,797,659
2012	250,673	1,737,726	2,829,608	4,818,006	306,470	2,184,113	5,496,505	7,987,088
2013	221,177	1,676,127	2,601,124	4,498,427	262,755	2,075,520	4,418,616	6,756,892
2014	249,371	1,629,858	2,346,243	4,225,473	287,866	1,941,016	2,828,942	5,057,824
2015	157,571	1,594,249	2,070,475	3,822,295	176,747	1,869,110	1,455,156	3,501,013
2016	11,523	1,560,600	1,783,636	3,355,759	12,559	1,847,492	704,641	2,564,692
2017	9397	1,487,944	1,464,042	2,961,383	9952	1,783,839	378,664	2,172,455

Note: Total for lower bound is 34,138,334, total for higher bound is 43,708,839.

Table B7
Carbon sequestration values over time.

	With policy Lower			Higher			Policy contribution Lower			Higher		
Year	Date	Canary	Total	Date	Canary	Total	Date	Canary	Total	Date	Canary	Total
2005	337	408.8	746	1643	1991.4	3634	0	0	0	0	0	0
2006	328	395.1	723	1551	1869.5	3421	1	12	13	2	16	18
2007	319	371.8	691	1467	1739.1	3207	1	17	18	5	29	36
2008	310	345.8	656	1385	1597.3	2983	2	21	23	7	45	52
2009	301	318.3	620	1311	1443.8	2755	2	27	29	11	73	84
2010	293	289.2	582	1241	1265.3	2506	2	33	35	14	111	126
2011	284	259.1	543	1173	1054.4	2227	2	41	43	17	163	180
2012	277	229.0	506	1109	811.1	1920	3	50	54	19	228	248
2013	270	199.1	469	1046	552.2	1598	4	61	65	17	287	308
2014	262	169.7	432	987	320.4	1307	4	70	74	17	289	312
2015	255	141.9	397	932	161.0	1093	4	77	82	16	175	203
2016	248	116.5	364	877	79.5	956	3	80	85	9	60	89
2017	241	94.1	335	824	46.6	870	3	78	83	1	107	141
Total			7064			28,477			604			1797

Appendix C. Regression results

Table C1
Regression results for date palms.

	Dependent variable: Distribution of date palms (1)	(2)	(3)
Industry contribution to GDP (%)	− 158.000***		− 120.000*
	(48.000)		(60.600)
Distance to the coastline	− 0.655***	− 0.558***	− 0.742***
	(0.174)	(0.170)	(0.192)
Services contribution to GDP (%)		157.000***	
		(48.400)	
Construction contribution to GDP (%)			376.000
			(358.000)
Constant	92.400***	− 52.600*	60.200*
	(18.000)	(31.600)	(35.600)
Observations	73	73	73
R^2	0.226	0.222	0.238
Adjusted R^2	0.204	0.200	0.205
Bayesian Inf. Crit.	813.000	814.000	816.000
Residual Std. Error	57.700	57.900	57.700
	(df = 70)	(df = 70)	(df = 69)
F Statistic	10.200***	9.970***	7.180***
	(df = 2; 70)	(df = 2; 70)	(df = 3; 69)

*p < 0.1
**p < 0.05
***p < 0.01.

Table C2
Regression results for canary palms.

	Dependent variable:		
	Distribution of canary palms		
	(1)	(2)	(3)
Distance to the coastline	− 0.428***	− 0.493***	− 0.377***
	(0.094)	(0.098)	(0.102)
Population			0.0004
			(0.0003)
Services contribution to GDP (%)	117.000***		104.000***
	(26.800)		(28.600)
Industry contribution to GDP (%)		− 109.000***	
		(27.100)	
Constant	− 36.400**	69.100***	− 33.600*
	(17.500)	(10.200)	(17.600)
Observations	73	73	73
R^2	0.347	0.325	0.362
Adjusted R^2	0.329	0.305	0.334
Bayesian Inf. Crit.	728.000	730.000	730.000
Residual Std. error	32.100	32.600	31.900
	(df = 70)	(df = 70)	(df = 69)
F statistic	18.600***	16.800***	13.100***
	(df = 2; 70)	(df = 2; 70)	(df = 3; 69)

*p < 0.1.
**p < 0.05.
***p < 0.01.

Table C3
Regression results for property values.

	Dependent variable:		
	Asking price		
	(1)	(2)	(3)
Bath number	50,097.000***	52,095.000***	51,538.000***
	(5140.000)	(5092.000)	(5170.000)
Built area	302.000***	314.000***	277.000***
	(41.700)	(41.500)	(41.600)
Garden	32,441.000***	37,402.000***	
	(7389.000)	(7145.000)	
Parking	19,229.000**		27,976.000***
	(7554.000)		(7344.000)
Balcony	1789.000	2982.000	5518.000
	(13,402.000)	(13,426.000)	(13,480.000)
Wardrobe	43,247.000***	46,131.000***	48,603.000***
	(7063.000)	(6988.000)	(7011.000)
Plot of land	27.200***	28.200***	34.900***
	(7.230)	(7.240)	(7.070)
Age	− 197.000***	− 193.000***	− 222.000***
	(65.600)	(65.700)	(65.800)
Constant	16,327.000	17,353.000	23,835.000**
	(11,943.000)	(11,964.000)	(11,912.000)
Observations	1169	1169	1169
R^2	0.312	0.308	0.301
Adjusted R^2	0.307	0.304	0.296
Bayesian Inf. Crit.	30,553.000	30,552.000	30,565.000
Residual Std. Error	111,591.000 (df = 1160)	111,854.000 (df = 1161)	112,466.000 (df = 1161)
F Statistic	65.700*** (df = 8; 1160)	73.900*** (df = 7; 1161)	71.300*** (df = 7; 1161)

*p < 0.1.
**p < 0.05.
***p < 0.01.

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